LCA Methodology with Case Study

Eutrophication of Aquatic Ecosystems

A New Method for Calculating the Potential Contributions of Nitrogen and Phosphorus

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DOI: http://dx.doi.org/10.1065/lca2004.02.145

Abstract

Aim, Scope and Background. Aquatic eutrophication is a widespread problem in inland and coastal waters around the world and it should therefore be one of the impact categories to be considered in LCA studies of products and services. In LCAs there are several impact assessment methods to determine characterisation factors for eutrophying nutrients, but few methods have been developed to model fate and spatial aspects. One such method was developed as part of an LCA application of the Finnish forest industry. The aim of this study was to present this characterisation method in which the potential contributions of nitrogen and phosphorus to eutrophication of aquatic ecosystems are calculated. The use of the method was demonstrated by producing site/sector-specific characterisation factors and by constructing a reference value of aquatic eutrophication for Finland. A discussion of sensitivity and uncertainty aspects related to input data is also presented.

Methods. The potential contribution to eutrophication from a product system is calculated as a result of the nutrient inputs causing increased production of biomass within aquatic systems. Accordingly, direct nitrogen and phosphorus emissions as well as nitrogen and phosphorus deposition into the watercourses can be included. In the method, characterisation factors for nitrogen and phosphorus emissions are generated by multiplying commonly used equivalency factors by transport and effect factors. Transport and effect factors of the nutrient sources are case-specific and can be determined for each substance individually on the basis of scientific models, empirical data or expert judgements. In this paper, transport and effect factors are determined for different sectors (forest industry, field cultivation, fish farming, etc.) in Finland. In addition, temporal aspects can be taken account of in the model by coefficients representing the proportion of nutrient load in the productive period of the total annual load. The model uncertainty was studied by using three different scenarios based on different input data and assumptions. Uncertainties within the input data were assessed as ranges and the effects of input data uncertainty on the results were studied by varying maximum and minimum values of each input variable in the same time.

Results and Discussion. The characterisation method developed was applied to provide characterisation factors of aquatic eutrophication for different sectors in Finland. The magnitude of these sector-specific characterisation factors varied greatly. For this reason, the method results were easily differentiated from those commonly used, site-generic characterisation factors produced in LCIAs. Furthermore, reference values of aquatic eutrophication for normalisation purposes in LCIA were generated on the basis of sector-specific nutrient emissions and characterisation factors. Different scenarios produce alternative pictures of the contributions of various sectors to aquatic eutrophication. By examining the scores of different sectors, it can be concluded that uncertainties in the emission estimates of field cultivation and fate of nitrogen originating from deposition have the greatest effect on the results. In the average scenario, the uncertainty range for the reference value was estimated to be ± 40% due to all uncertainties in input variables.

Conclusions. The results of the work reveal the importance of site-specific characterisation approach in the context of aquatic eutrophication. Furthermore, differentiation of nutrient forms in various sectors means that the question of determination of characterisation factors is also related to sector-specific issues. The method demonstrated is flexible, and it can be applied to geographical areas rather than to Finland as a whole. The use of sensitivity and uncertainty analysis is recommended for the interpretation of results, because there is no empirical test applicable for evaluation of the validity of results. In order to reduce uncertainty in results, further research is needed, in particular on the roles of different nutrient forms as sources for aquatic biota, on spatial differentiation of nitrogen and phosphorus as production-limiting factors, and on fate of nitrogen in catchments.

Outlook. The weakness of the method is related to the accessibility of input data, restricting to construct the characterisation model of aquatic eutrophication, for example, on the European level. However, it seems that legislative requirements of the European Union to study 'target nutrient loads' of aquatic eutrophication in the catchments of each Member State can improve the situation of the accessibility of input data in the near future.

Keywords: Aquatic ecosystems; eutrophication; life cycle impact assessment (LCIA); nitrogen; nutrients; phosphorus; sensitivity analysis

Introduction

Eutrophication of aquatic ecosystems can be defined as the state of a water body in which the production and accumulation of algae and higher aquatic plants have increased excessively due to increased input of plant nutrients (e.g. Rast and Ryding 1988). These changes can result in undesirable effects on water quality and on the biological populations of the water body. Bad taste and odour associated with excessive algal growth limit the water body's value as a source of drinking water and as a recreational area. Decomposition of dead algal biomass reduces the benthic dissolved oxygen level.

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Aquatic eutrophication is a widespread problem in inland and coastal waters around the world and it should therefore be one of the impact categories to be considered in life cycle assessment (LCA) (e.g. Udo de Haes et al. 1999). LCA is a method for analysing and assessing the environmental impacts of a material, product or service throughout its entire life cycle starting from raw material extraction and ending with final disposal.

An LCA study is initiated by goal and scope definition, in which the problem and the aims of the study are defined. In inventory analysis, data about environmental interventions (emissions, resource extractions and land use) during the life cycle of the product or service is collected. Usually the results of the inventory are difficult to interpret. In such cases, impact assessment methods can be helpful in order to obtain a better view of environmental impacts caused by the product or service.

The first scope of life cycle impact assessment (LCIA) is to estimate the potential contribution of different interventions (emissions, resource extractions and land use) to different impact categories (aquatic eutrophication, acidification, tropospheric ozone formation, etc.) and to sum the amounts of interventions into a single number within each impact category. In practice, this so-called characterisation is carried out by multiplying amounts of interventions and characterisation factors. In LCAs there are several impact assessment methods to determine characterisation factors for eutrophying nutrients, but few attempts have been made to model fate and spatial aspects (Finnveden and Potting 1999, Potting et al. 2002). One such method was developed as part of an LCA application of the Finnish forest industry (Seppälä et al. 1998). The characterisation method has been used in the context of an impact assessment method, called DAIA (Decision Analysis Impact Assessment) (Seppälä 1997, 1999).

The basic version of DAIA includes all phases of LCIA (ISO 2000): selection of impact categories, classification (assignment of the inventory data to the impact categories), characterisation (quantification of the contributions of the inventory data to the chosen impacts in terms of the category indicators), normalisation (the characterized results, i.e. category indicator results, are divided by reference values) and weighting (conversion of indicator results into a single index by using numerical factors based on value choice). In DAIA the calculation is conducted according to multiattribute value theory (MAVT) (e.g. Keeney and Raiffa 1976, von Winterfeldt and Edwards 1986, French 1988), which offers rational rules to aggregate different impacts into a single score. According to MAVT, normalisation is needed before weighting (Seppälä and Hämäläinen 2001). Thus, reference values are needed for each impact category if the aim is to produce total impact values caused by the product or service.

The aim of this study was to present a characterisation method for calculation of the potential contributions of nitrogen and phosphorus to eutrophication of aquatic ecosystems used in DAIA. The use of the method was demonstrated by producing sector-specific characterisation factors and by constructing a reference value of aquatic eutrophication for Finland. In addition, a discussion of sensitivity and uncertainty aspects related to input data is presented.

1 Method

The objective of the eutrophication impact method in LCIA is to identify differences between alternative systems with regard to potential eutrophication of aquatic ecosystems. These alternative systems can be any anthropogenic nutrient generation systems of products. The potential contribution to eutrophication from system a is calculated as a result of the nutrient inputs causing increased production of biomass in aquatic systems. Accordingly, direct nitrogen N(W) and phosphorus P(W) emissions, as well as nitrogen N(A) and phosphorus P(A) deposition into the watercourses, can be included. For this model, the impact value of eutrophication caused by system a, denoted $I_{Eu}(a)$ and called the category indicator result of eutrophication, is given by (Seppälä 1997, 1999)

$$I_{Eu}(a) = \sum_{j=1}^{n} x_j(a) * Eqv_j \tag{1}$$

where

$x_j(a)$ = effective input (rating) of substance j from system a
Eqv_j = equivalency factor of substance j

The philosophy behind the equivalency factor, Eqv_j, is to aggregate deposition of nitrogen and phosphorus as well as emissions of nitrogen and phosphorus into the waterways. Determination of Eqv_j is based on the amount of phytoplankton which the available nitrogen and phosphorus will give rise through photosynthesis in the photic zone of aquatic ecosystems. It is essential, when making these calculations, to determine the N:P ratios that may occur in phytoplankton in nature. This ratio differs between species, changes with time and depends on the dominating species in the photic zone. The ratio also depends on the nutrient status of the ecosystem. A ratio that has been used in several studies is 106:16:1 (C:N:P) suggested by Redfield et al. (1963). Thus, 1 mol of nitrogen emitted to an aquatic system will give rise to 6.6 mol organic carbon in biomass and, in the case of phosphorus, the corresponding amount is as much as 106 mol. On the basis of this ratio, equivalency factors for nitrogen and phosphorus can be expressed as PO_4-equivalents (**Table 1**). This approach is commonly used in LCA studies (e.g. Heijungs et al. 1992, Lindfors et al. 1995) in which the equivalency factors are usually called characterisation factors and the ratings correspond to emissions of nutrients.

Table 1: Equivalency factors for substances causing eutrophication in aquatic ecosystems (Heijungs et al. 1992)

Substance	Equivalency factor (as PO_4 equivalent)
N to air	0.42
NO_x to air	0.13
NH_3 to air	0.35
N to water	0.42
NO_3 to water	0.10
NH_4 to water	0.33
P to water	3.06
PO_4^{3-} to water	1

According to Equation 1 the effective nutrient inputs of each system must be assessed for each substance j as measurement levels, denoted $x_j(a)$. This 'rating' represents the quantity of an emission that affects photosynthesis in aquatic ecosystems impacted by cultural eutrophication. The response caused by emissions depends, among other things, on the locations of the emission sources. Spatial aspects of nutrient sources can be taken into account via the rating in the method. The rating can be assessed by the following equation (Seppälä 1997, 1999):

$$x_j(a) = \eta_j(a) * \mu_j(a) * E_j(a) \qquad (2)$$

where

$\eta_j(a)$ = transport factor of substance j ($0 \leq \eta_j \leq 1$)
$\mu_j(a)$ = effect factor of substance j ($0 \leq \mu_j \leq 1$)
$E_j(a)$ = emitted amount of substance j due to system a

The transport factor η_j indicates which part of a given water area may be affected by eutrophying emission E_j. The effect factor μ_j indicates what quantity of the transported substance j causes increased production of biomass in the area. Both factors can be determined for each substance individually on the basis of scientific models, empirical data or expert judgements. In fact the transport factors can be calculated by models simulating the fate of a substance in the environment. Definition of the transport and effect factors depends on the type of problem (Seppälä 1997, 1999).

The higher the value score $I_{Eu}(a)$, the more undesirable is the system. Thus, magnitudes of $I_{Eu}(a)$ can be used to establish a ranking that indicates the decision maker's preferences for the alternative systems.

In summary, Equation 1 can be written according to the traditional terminology of LCIA as follows:

$$I_{Eu}(a) = \sum_{j=1}^{n} C_j(a) * E_j(a) \qquad (3)$$

where

$C_j(a)$ is a characterisation factor of substance j caused by system a under eutrophication.

Thus,

$$C_j(a) = \eta_j(a) * \mu_j(a) * Eqv_j \qquad (4)$$

Furthermore, the impact category indicator in the method is algal growth.

The output of Equation 1 depends critically on data input, and these data should therefore be examined critically. It is essential to check the sensitivity of the value score $I_{Eu}(a)$ against different views about the uncertainties associated with the assessment model. Sensitivity and uncertainty analysis allows the identification of critical judgements and the identification of competitors of alternative a.

For this investigation, it is convenient to think of Equation 1 as producing an output, such as a value set of system a, that is a function of several input variables $\eta_j(a)$, $\mu_j(a)$, $E_j(a)$ and Eqv_j with inherent uncertainties. In practice, understanding of the correct values for these input variables is never perfect. Uncertainty in variables can be described by probability density functions. Uncertainty analysis involves determination of the variation in the output function on the basis of collective variability of model inputs. The uncertainty analysis can be carried out by a Monte Carlo approach (see e.g. Environmental Resources Limited 1985). If uncertainties cannot easily be accounted for by Monte Carlo simulation, easy-to-use and transparent methods such as sensitivity analysis can be applied by varying the intervals of individual variables to ensure the stability of the results.

2 Data and assumptions for the calculation of source-dependent characterisation factors and a reference value for Finland

2.1 Background

A starting point for a site-specific impact assessment in LCIA is that the applied characterisation method can also produce impact indicator results of the product as well as normalisation reference values on the same basis. A normalisation reference value can be the total characterised impact indicator result calculated on the basis of an inventory of all of society's activities in some given area and over a reference period of time (Consoli et al. 1993, Wenzel et al. 1997). In practice, the time frame is always one year in LCA applications, but areas typically vary.

In this study, different sectors in Finland covering all the important sources of eutrophying emissions were chosen as a basis for calculation of the normalisation reference value of aquatic eutrophication. The geographical area was restricted to Finland for practical reasons. Finland is a complex area from the point of view of aquatic eutrophication, offering a good case area for demonstrating the method.

The area of all inland waters in Finland is 33,522 km^2, i.e. 9.9% of the total area of the country (Statistics Finland 1996). Despite their large number and total surface area, Finnish lakes are sensitive to changes because of their low buffering capacity and low nutrient concentrations, their small volume and also the long ice-cover period (up to 8 months in Lapland). Most of the Finnish watercourses discharge to the Baltic Sea (**Fig. 1**).

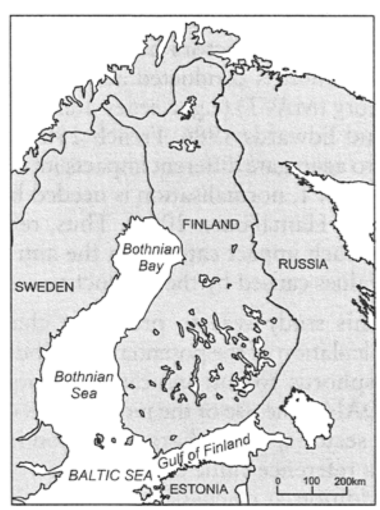

Fig. 1: General map of the Finnish coastal and inland waters

Eutrophication of lakes, rivers and coastal waters is considered one of the main environmental problems in Finland (Wahlström et al. 1996). Water quality in most Finnish watercourses is good or excellent, but algal blooms are a common problem in the intensively cultivated regions of southern and western Finland (Kauppi et al. 1993). As much as 30% of the total length of Finnish rivers is of poor quality, although about 45% is still excellent or good. Excessive nutrient inputs to the Baltic Sea also give rise to serious concern. Since the early 1900s, the status of the Baltic Sea has changed from an oligotrophic clear-water to a highly eutrophic sea (Larson et al. 1985). During the twentieth century, phosphorus inputs to the Baltic Sea increased eight-fold, while nitrogen loads increased four-fold (Helsinki Commission 1996, Elmgren 2001). High concentrations of phosphorus and nitrogen have caused algal blooms, especially of blue-green algae.

Most of the nutrients in Finnish inland waters originate from cultivated areas. With regard to the state of Finnish coastal waters, the role of riverine loading from Finland and from the St. Petersburg region in Russia is particularly important (e.g. Helsinki Commission 2001, Kauppila and Bäck 2001). As a rule, phosphorus discharges from point sources have decreased markedly, but nitrogen discharges only slightly during recent years.

2.2 Emissions

The activities causing eutrophication are divided in the context of waterborne nitrogen and phosphorus emissions into point and diffuse sources (**Table 2**). In addition, depositions of nitrogen and phosphorus are taken into account (see Section 2.3).

Estimation of average emissions from point sources to water in the first LCIA application (Seppälä et al. 1998) was based on official statistics (Repo and Hämäläinen 1996) representing the following years: industry 1995, communities 1990–1993 and fish farms 1993. The average fluxes of total N and P from field cultivation are estimated from the study by Rekolainen (1993). These fluxes vary considerably from year to year according to hydrological conditions (e.g. Rekolainen 1993, Pitkänen 1994). The load estimates of field cultivation represent average load levels in the early 1990s. The same applies to the estimation of other diffuse sources (scattered population, forestry, peat production) and semi-diffuse sources (fur farms and horticulture). Their load estimates were based on material gathered during preparation of the national programme to protect the aquatic environment (Ministry of the Environment 1998).

In an LCA study of Finnish rainbow trout cultivation (Seppälä et al. 2001) the average nutrient emissions from Finnish point sources were based on an official updated estimation performed by Silvo et al. (2002). These values represented the situation in the year 2000.

2.3 Transport factors

Biomass growth may be limited by different nutrients in different waterways. Earlier it was assumed that phosphorus was generally the limiting nutrient in Finnish inland waters (Pietiläinen 1997). Furthermore, it was assumed that only direct phosphorus P(W) emissions into waters cause eutrophication of these inland waters. However, in the Baltic Sea, excluding Bothnia Bay (the northernmost part of the Baltic Sea), nitrogen is the main cause of increased production of biomass (Tamminen and Kivi 1996).

In the determination of the transport factors $\eta_{P\,(A)}$ and $\eta_{N\,(A)}$ for the different sectors A, the above-mentioned aspects were taken into account in scenario 1 (**Table 3**), which was used in the early LCA applications (see Seppälä 1997, 1999). Consequently, the direct nitrogen load into the watercourses within the catchments of this sea area was also included in the calculations. The fact that phosphorus also causes increased production of biomass along the Finnish coast was considered (Pitkänen et al. 1994) and it was assumed that all domestic phosphorus emissions to the Baltic Sea will cause an increased production of biomass.

Table 3: Estimated transport factors of nitrogen η_N and phosphorus η_P for the Finnish load sources in scenario 1

Source	η_N	η_P
Pulp and paper industry	0.35	1
Other industry	0.25	1
Communities	0.45	1
Fish farms	0.80	1
Fur farms	0.10	1
Horticulture	0.60	1
Scattered population	0.48	1
Field cultivation	0.50	1
Livestock	0.30	1
Forestry	0.25	1
Peat production	0.15	1

Table 2: Waterborne nitrogen and phosphorus fluxes into waters from Finnish point and diffuse sources in the early 1990s and in 2000

Source	Average N into water (t/a)		Average P into water (t/a)	
	Early 1990s	2000	Early 1990s	2000
Pulp and paper industry	3,157	2,846	320	224
Other industry	1,745	1,101	37	26
Communities	14,500	12,261	270	259
Fish farms	1,600	950	290	122
Fur farms	430	430	45	45
Horticulture	345	345	7	7
Scattered population	2,700	2,730	415	410
Field cultivation	30,000	38,000	3,000	2,650
Livestock	2,900	1,900	300	250
Forestry	3,330	4,180	340	240
Peat production	1,100	1,100	50	50

Table 4: Airborne nutrient fluxes and their transport factors

Finnish source	Load (t/a)		Transport factor η	
	Early 1990s	2000	Scenario 1	Scenario 2
Emissions				
– NO_x to air	76,000	72,040	0.06	0.17
– NH_3 to air	34,000	29,000	0.07	0.16
Deposition of phosphorus to Finnish inland waters and the coastal zone	350	410	1	1

In addition, in the determination of the transport factors $\eta_{P\,(A)}$ and $\eta_{N\,(A)}$ for the different sectors, the geographical locations of the load sources in each sector were determined. According to these data the final coefficients were determined by expert judgement. Expert judgement is here defined as judgements made by those with expertise or knowledge in the area.

In scenario 1, estimated deposition of nitrogen to water was based on domestic oxidised nitrogen (NO_x) and reduced nitrogen (NH_y) emissions, and the calculations of coefficients for atmospheric loads η_{NOx} and η_{NHy} obtained from the results of the EMEP model (Co-operative Programme for Monitoring and Evaluation of the Long Range Transmission of Air Pollutants in Europe) published by Barret et al. (1995). According to the average data from the years 1985–1994, approximately 6 percent of the annual domestic oxidized nitrogen (NO_x) emissions were deposited into the sea area described above and into the North Sea. The corresponding value for reduced nitrogen (NH_y) emissions was assessed to be 7% (**Table 4**). The variations in deposition are not known exactly because of the aggregated data in the EMEP calculations.

One clear weakness in scenario 1 is that the transportation of nitrogen from inland to sea areas may be underestimated. This conclusion is supported by the recent study by Lepistö et al. (2001) concerning the nitrogen export budget in a northern river basin in Finland. Scenario 1 also underestimates the role of nitrogen deposition on the sea area. Although eutrophication is not a problem out in the Atlantic Ocean, the deposition increases the background concentration of the sea water.

For the reasons described above, scenario 2 includes different transport factors which may overestimate the role of nitrogen as a eutrophying emission. The starting point is that all nitrogen reaching the waters is taken into account. It is assumed that 5 percent of nitrogen deposited on soil leaches to waters in Northern Europe (c.f. Lepistö et al. 2001), whereas in Central Europe the corresponding proportion is 30%. Scenario 2 also takes into account the amount of deposition on the sea area of the EMEP-model outside the Baltic Sea and the North Sea caused by human activities in Finland. Only half of this amount is included as 'effective' nitrogen.

Assessment of phosphorus deposition to water in the data concerning the early 1990s was based on the data of average phosphorus concentrations of rainwater during the years 1992–1993 (Järvinen and Vänni 1994a, Järvinen and Vänni 1994b), average precipitation in Finland, areas of inland waters and the length of the coastal shore (Statistics Finland 1996). Only a 10 km wide zone was taken into account in the determination of the coastal area. This zone is quite arbitrary, based on the assumptions that the contribution of phosphorus to eutrophication of the pelagic area in the Baltic Sea is insignificant, and that phosphorus in rainwater samples mainly originates from wind-driven particles from agricultural sources which are not transported far from the shore. In addition, spatial variation of phosphorus concentrations in the measured rainwater were taken into account in the calculation and the effects of deposition from nature and of emissions from abroad were estimated by eliminating background deposition (2–5 P mg/m^2). Due to all the above-mentioned assumptions, the estimation of the deposition presented in Table 4 is very approximate and can be considered as 'effective deposition to water' without the need for transformation by a transport factor.

The value of phosphorus deposition to water in 2000 was obtained from the official assessment of the Finnish Environment Institute (2001).

2.4 Effect factors

The biological availability of total N and P released from various anthropogenic sources varies considerably. An average phosphorus emission from agriculture, for example, may cause less eutrophication than the same amount of phosphorus from a pulp mill in the same watercourse, because of the different chemical forms of nutrients in the emissions. Biological availability of a nutrient is here considered as the sum of nutrient forms that can be transformed into a form available to algae. In the model , biological availability of P and N is a basis for determination of effect factors ($\mu_i(a)$).

In the first aquatic eutrophication assessment model of DAIA (Seppälä 1997, 1999) (called an old model in **Table 5**) the following data and assumptions were used for determination of effect factors of phosphorus. A study of P discharges from the Finnish forest industry (Priha 1994) suggests a bioavailability of 80% for P inputs. In municipal discharges the corresponding P value is estimated to be 40% (Ekholm et al. 1994). These figures were used for effect factors of P discharges from communities and scattered populations in this study. According to Ekholm (1998), it can be estimated

Table 5: Estimated effect factors of nitrogen (μ_N) and phosphorus (μ_P) for the Finnish load sources

Source	μ_N	μ_P	
		Old model	New model
Pulp and paper industry	0.50	0.80	0.30
Other industry	0.90	0.50	0.50
Communities	0.90	0.40	0.40
Fish farms	0.90	0.90	0.30
Fur farms	0.80	0.80	0.80
Horticulture	0.70	0.30	0.30
Scattered population	0.80	0.40	0.80
Field cultivation	0.70	0.30	0.30
Livestock	0.60	0.80	0.80
Forestry	0.20	0.30	0.30
Peat production	0.20	0.30	0.30
Deposition originating from			
– NO_x to air	1	–	–
– NH_3 to air	1	–	–
– other		0.50	0.50

that 28% of the diffuse total P load originating from agriculture is potentially available to algae. The directly bioavailable P inputs from forested areas are probably only 2.5% of the total P losses (Pitkänen 1994). This assessment was a basis for effect factors of P from forestry and peat production. The determination of bioavailability of total P from livestock was based on the proportion of dissolved reactive P in the total P in urine and manure (Kemppainen 1992). Regarding other sources, the average bioavailability of P was obtained by expert judgements gathered during preparation of the new Water Protection Targets to 2005 (Marttunen 1998). Experts working with eutrophication issues at the Finnish Environment Institute gave their assessments on the basis of their knowledge of phosphorus forms in each discharge. In practice, the assessments corresponded to the proportion of dissolved orthophosphate (and other P forms immediately available to algae) of the total P in discharges.

The bioavailability of total N for the different sources was based on the proportion of inorganic N of the total N in discharges. Estimation of the inorganic N in discharges from livestock was obtained from the literature value of the composition of manure represented by Kemppainen (1992). According to Lepistö et al. (1995) the average proportion of inorganic N of the total N in discharges of small forested catchments is approximately 20%. This was a basis for estimation of the proportion of inorganic N in discharges from forestry and peat production. The average estimations for the other sources were obtained from the national discharge monitoring database managed by the environmental administration in Finland.

In a current version of DAIA, the estimations of bioavailable P of many sectors were changed from the early values due to the recent study of Ekholm and Krogerus (2003).

2.5 Temporal aspects

The above-mentioned values for transport and effect factors were used in the basic calculations, called scenarios 1 and 2. However, these scenarios do not include temporal aspects related to aquatic eutrophication. In scenario 3, temporal variation of biological availability caused by the environmental conditions was taken into account in the determination of effect factors ($\mu_j(a)$). Furthermore, transport factors in scenario 3 represent the average values calculated by the transport factors used in scenarios 1 and 2.

To add temporal aspects to the model, the effect factors presented in Table 5 were multiplied by coefficients representing the proportion of load in the productive period of the total annual load (**Table 6**), i.e. in scenario 3 the effect factor is defined as

$$\mu_j(a) = \mu^B{}_j(a) * load^S{}_j(a) / load^A{}_j(a) \tag{5}$$

where

$\mu^B{}_j(a)$ = effect factor of substance j on the basis of biological availability (presented in Table 5 and used in scenarios 1 and 2 as effect factor)

$load^S{}_j(a)$ = emitted amount of substance j due to source a in the productive period

$load^A{}_j(a)$ = emitted annual amount of substance j due to source a

In the calculations of scenario 3, the productive period is May–September in southern and central Finland and July–August in the north. Furthermore, in the context of domestic nitrogen air emissions, the productive period for nitrogen deposition below latitude 55 degrees north is considered to be April–October.

Table 6: Nitrogen and phosphorus loads to waters in the productive season (load ^S_j) as proportions of the total annual load (load ^A_j)

Finnish source	$load^S_j / load^A_j$
Pulp and paper industry	0.40
Other industry	0.40
Communities	0.40
Fish farms	0.85
Fur farms	0.40
Horticulture	0.45
Scattered population	0.45
Field cultivation	0.35
Livestock	0.15
Forestry	0.35
Peat production	0.35
Deposition originating from	
– NO_x to air	0.50
– NH_3 to air	0.50
– other	0.45

2.6 Uncertainty factors

In this work, scenario analysis was a starting point to study uncertainty aspects of the method. The differentiations of the results calculated by scenarios 1, 2 and 3 reveal the model uncertainty. In addition, effects of uncertainties of input data were assessed in the case study of the Finnish forest industry (Seppälä et al. 1998), where model input values were specified as uniform distribution with minimum and maximum values. These ranges were specified by expert judgements. The uncertainty interval of the average loads from point sources was ± 10% and from diffuse sources ± 30%. The uncertainty intervals for transport factors $\eta_{N(A)}$ were judged to be ± 0.05. Exceptions were horticulture, field cultivation and depositions of NO_x and NH_3 in which uncertainty intervals of transport factors of nitrogen were ± 0.1, ± 0.1, ± 0.01 and ± 0.01, respectively. The uncertainty intervals of effect factors were determined to be ± 0.1 with the exception of fish farms (± 0.05) and other industry (± 0.05). The uncertainty interval for nitrogen and phosphorus load into waters in the productive season (*load* ^S_j) as a proportion of the total annual load (*load* ^A_j) was assessed to be ± 0.05 for each sector. In addition, it was assumed that the range of the N:P ratio can be between 13:1 and 19:1. This is due to the fact that the N:P ratio of phytoplankton is known to vary in different water bodies (e.g. Samuelsson 1993, Crouzet et al. 1999).

3 Results and Discussion

In this work, source-dependent characterisation factors of aquatic eutrophication for each source sector were calculated according to the different scenario data presented in Sections 2.2–2.5 (**Table 7**). The traditional characterisation factors of eutrophication used in LCIA are 0.42 t PO_4/ t N and 3.06 t PO_4/ t P (see Table 1). The values presented in Table 7 are clearly smaller than these traditional factors. However, the magnitude of characterisation factors is not important for the purpose of LCIA. There is no sense in comparing values of different scenarios with each other. Only the relative differentiations between characterisation factors of different sectors within each scenario are important. In characterisation, nutrient emissions of each sector are multiplied by the corresponding characterisation factors. For example, according to scenario 1, one kg of nitrogen emission into water from a fish farm causes a 15–30-fold response in aquatic eutrophication compared to one kg of nitrogen emissions into air (see Table 7). This kind of differentiation can easily alter conclusions about the contributions

Table 7: Estimated average characterisation factors for nitrogen (N) and phosphorus (P) emissions into water from Finnish sources on the basis of different assumptions (scenarios 1–3)

Source	Scenario 1		Scenario 2		Scenario 3	
	N	P	N	P	N	P
Pulp and paper industry	0.07	0.92	0.21	0.92	0.06	0.37
Other industry	0.09	1.53	0.38	1.53	0.09	0.61
Communities	0.17	1.22	0.38	1.22	0.11	0.49
Fish farms	0.30	0.92	0.38	0.92	0.29	0.78
Fur farms	0.03	2.45	0.34	2.45	0.07	0.98
Horticulture	0.18	0.92	0.29	0.92	0.11	0.41
Scattered population	0.16	2.45	0.34	2.45	0.11	1.10
Field cultivation	0.15	0.92	0.29	0.92	0.08	0.32
Livestock	0.08	2.45	0.25	2.45	0.02	0.37
Forestry	0.02	0.92	0.08	0.92	0.02	0.32
Peat production	0.01	0.92	0.08	0.92	0.02	0.32
Deposition originating from						
– No_x to air	0.03		0.07		0.02	
– NH_3 to air	0.03		0.07		0.02	
– other		1.53		1.53		0.69

of different emission sources of a studied product system to aquatic eutrophication compared to the results produced by the traditional characterisation factors.

Spatial differentiation of nitrogen and phosphorus as production-limiting factors is a basis for determination of the transport factor. In scenario 1, it was assumed that nitrogen does not cause an increased production of biomass in inland waters, whereas scenario 2 assumed that all nitrogen reaching inland waters causes eutrophication effects. Neither scenario handles nitrogen in the right way. In general, it can be said that nitrogen is not a limiting nutrient in inland waters. However, this does not mean that nitrogen released to inland waters has no effect on aquatic eutrophication. The question, 'how much of gross N loads in the catchments reach the Baltic Sea?', is still lacking an exact answer. This concerns both the direct and indirect loads to water bodies. The indirect load also includes the question concerning what is the amount of N deposition passing through the catchment soils to water bodies. There is a need for further research in the field of retention. In addition, the role of nitrogen as a eutrophying nutrient in inland waters is still under investigation (e.g. Pietiläinen and Räike 1999, Crouzet et al. 1999).

Different scenarios produce alternative pictures of the contributions of various sectors to aquatic eutrophication. The differentiation of results obtained from scenarios 1 and 2 is most important in the context of nitrogen deposition. The differentiation is great enough to have effects on normalisation values in LCIA. For this reason, the question rises: which is the best scenario to provide conclusions? As described above, scenario 1 probably underestimates the role of nitrogen compared with phosphorus, whereas scenario 2 overestimates the role of nitrogen. One solution is to calculate aquatic eutrophication indicator results using both scenarios and to study whether their results have different effects on the conclusions. This approach was used in an LCA case study of Finnish rainbow trout cultivation, in which aquatic eutrophication was shown to be the most significant impact category caused by all life cycle stages concerning the rainbow trout cultivation (Seppälä et. al. 2001).

Scenario 3 with its temporal aspects offers an alternative approach to interpretation compared with scenarios 1 and 2. However, scenario 3 cannot be said to be better than scenarios 1 and 2. The assumption concerning the productive period is very approximate without exact scientific data. Although harmful effects of eutrophication in Finnish coastal and inland waters occur in the late summer period, there may be many places where indirect 'old' loads cause the effects because the 'non-effective' nutrient load can be translocated to the water phase in the summer period. Furthermore, the importance of uncertainty effects on results can be studied by combining temporal variation (see Section 2.4) with scenarios 1 and 2 instead of using their average results as done in scenario 3.

The total value scores calculated as PO_4-equivalents reveal the contribution of each sector-specific nutrient flux to aquatic eutrophication. The contributions of emission changes and scenarios to the results of reference values in Finland are presented in **Table 8**. The total reference value decreased by 3% from the early 1990s to 2000 when calculated by scenario 1. The effect of corrections made for effect factors on the reference value of the year 2000 is not great: the reference value calculated by the old model is only 0.2%

Table 8: Estimated average nitrogen (N) and phosphorus (P) fluxes (t PO_4 eq/a) into water from Finnish sources on the basis of different assumptions (scenarios 1–3) and data

Data Source	Scenario 1			Scenario 2			Scenario 3		
	N	P	Sum	N	P	Sum	N	P	Sum
2000									
Pulp and paper industry	207	185	393	592	185	778	160	74	234
Other industry	111	44	156	445	44	489	111	18	129
Communities	2,076	305	2,381	4,613	305	4,918	1,338	122	1,460
Fish farms	308	115	423	386	115	500	295	98	392
Fur farms	14	110	125	144	110	255	32	44	76
Horticulture	57	30	87	95	30	125	34	14	48
Scattered population	440	1,004	1,444	917	1,004	1,921	305	452	757
Field cultivation	5,586	2,433	8,019	11,172	2,433	13,605	2,933	851	3,784
Livestock	144	612	756	479	612	1,091	47	92	138
Forestry	88	219	307	351	219	570	77	77	154
Peat production	14	46	60	92	46	138	19	16	35
Deposition originating from									
– NO_x to air	1,841		1,841	17,599		17,599	4,860		4,860
– NH_3 to air	811		811	2,243		2,243	764		764
– other		627	627		627	627		282	282
Sum	11,697	5,731	17,428	39,128	5,731	44,859	10,973	2,139	13,112
Early 1990s									
Sum	11,531	6,461	17,992	39,904	6,461	46,365	11,327	2,443	13,771
2000 Calculated by an old model									
Sum	11,697	5,767	17,465	39,128	5,767	44,896	10,973	2,232	13,205

Table 9: Estimated ranges for nitrogen (N) and phosphorus (P) fluxes (t PO_4 eq/a) into water from Finnish sources on the basis of the average scenario 1–2

Source	Nitrogen			Phosphorus			Total		
	Min	Max	Average	Min	Max	Average	Min	Max	Average
Pulp and paper industry	277	547	400	111	272	185	388	819	585
Other industry	217	348	278	32	59	44	249	407	322
Communities	2,843	4,017	3,345	206	419	305	3,049	4,436	3,649
Fish farms	287	414	347	69	168	115	356	582	462
Fur farms	46	121	79	67	161	110	114	283	190
Horticulture	43	120	76	14	53	30	57	172	106
Scattered population	393	1,040	679	615	1,468	1,004	1,008	2,508	1,682
Field cultivation	4,491	13,585	8,379	1,087	4,314	2,433	5,578	17,899	10,812
Livestock	175	490	311	375	895	612	549	1,385	923
Forestry	74	445	219	102	380	219	176	825	439
Peat production	18	108	53	21	80	46	39	188	99
Deposition originating from									
– NO_x to air	2,735	4,674	3,644				2,735	4,674	3,644
– NH_3 to air	928	1,859	1,354				928	1859	1,354
– other				452	828	627	452	828	627
Sum	12,526	27,770	19,164	3,151	9,096	5,731	15,677	36,866	24,895

greater compared with the reference value calculated by the new effect factors. However, it is clear that the changes can have significant effects on the sector-specific data.

The uncertainty data assessed in the case of the forest industry study (Section 2.6) can only offer a basis for approximate sensitivity/uncertainty analysis. The ranges are quite arbitrary and uniform distributions produce conservative results. In fact, the shapes of distributions are unknown due to a lack of data. In addition, the variables of phosphorus and nitrogen loads within each nutrient source may be correlated, which is difficult to assess. Thus, there is no basis to conduct uncertainty analysis by Monte Carlo simulation so that the results could be presented, for example as 95% confidence intervals for the value scores. For this reason, uncertainty analysis in this study was only conducted by using the ranges assessed for input variables in order to illustrate the magnitude of uncertainty related to the reference value of Finland. In the calculation, the total impacts of all uncertainties in input variables were taken into account in the same time. The starting point was the 'average scenario' in which the average values of transport factors were obtained from scenarios 1 and 2 (**Table 9**). By examining the scores of different sectors, it can be concluded that uncertainties in field cultivation and nitrogen deposition have the greatest effect on the results. In field cultivation, the uncertainty question is related to emission estimation, whereas the question is related to the fate of nitrogen in the catchments in deposition. When the uncertainties of temporal aspects (Sections 2.5 and 2.6) are added to the uncertainty analysis, the ranges of nutrients are increased (N: 4,436 – 12,262 t PO_4 eq/a and P: 1,023 – 3,851 t PO_4 eq/a).

The value scores of nitrogen and phosphorus are directly comparable with each other. In practice, it is possible to say that the contribution of nitrogen to eutrophication is greater than that of phosphorus when ranges are taken into account in the decision criteria. Furthermore, the calculation in which the uncertainty in equivalency factors is also taken into account increases the amount of the total P load (in the case of Table 9: 4,426-11,188 t PO_4 eq/a) compared with the amount of total N load, but the minimum value of nitrogen is still greater than the maximum value of phosphorus.

In the case study the focus was on Finnish emissions and the aim of the study was to determine a normalisation reference value for aquatic eutrophication caused by human activities in Finland. Furthermore, the sector-specific characterisation factors representing the average situation of each sector in Finland provide a basis for the calculations of eutrophication indicator results caused by product system *a* in Finland. However, the approach can be applied with a more detailed spatial scale in order to obtain more accurate results. For example, Finland can be divided into different catchment areas. For each catchment area an estimate of the total nutrient input from different sources is made. The catchment area can be further subdivided into sub-basins, generally on the basis of the drainage of smaller feeder streams into the main tributary. After establishing the sub-area boundaries, each important nutrient source and the location of its discharge are identified. The results of eutrophication models, limnology reports of monitoring and other sources of information can be used to define the transport and effect factors of the sources. The last step of the approach is to determine the total effective nutrient load for each substance and source on the basis of the catchment calculations. In future it will be worth applying the method to a simplified watershed model as presented e.g. by Reckhow et al. (1992).

In the case study, airborne nitrogen was taken into account as deposition. In the context of scenario 1, for example, the impact area of nitrogen deposition consists of the North Sea and the Baltic Sea. If nitrogen emissions from Finland were reduced the positive effect on eutrophication would mainly be observed in the sea area outside the Finnish coastal waters. The basis of the phosphorus deposition included in the case study differs from that of the nitrogen deposition. Its sources are unknown and, in practice, the possible eutrophication effects of domestic airborne phosphorus abroad were not taken into account in the calculations.

Therefore the nitrogen and phosphorus depositions in the case study are not directly comparable.

The determination of effect factors was based on the concept of biological availability of a nutrient, corresponding to the sum of directly available nutrient and nutrient that can be transformed into an available form for planktonic algae by naturally occurring physical, chemical and biological processes. As discussed by Boström et al. (1988) and Ekholm (1998), this definition is problematic, particularly with regard to the time perspective. There are problems in assessing amounts of available P and N due to e.g. sedimentation and the lag period after which benthic P and N may be translocated to the water phase. Especially nitrogen has a very complex cycle in waters. In practice, the determination of biological availability of nutrients in the case study was based on nutrient utilization by planktonic algae consuming the nutrients from a wide range of sources. In addition, the basis for determination of available nutrient for different sources varied significantly. On the other hand, aquatic bacteria and macrophytes may be able to utilise a wider variety of P forms than planktonic algae (Ekholm 1998). In order to reduce uncertainty in effect factors, further research is needed on the roles of different nutrient forms as sources for aquatic biota.

4 Recommendations and Outlook

The characterisation method for aquatic eutrophication due to Finnish nutrient emissions presented in this study is flexible, and it can be applied for LCIA of other geographical areas. In the characterisation method, nitrogen and phosphorus emissions are multiplied by equivalency, transport and effect factors. Transport and effect factors of the nutrient sources are case-specific and they can be determined for each substance individually on the basis of scientific models, empirical data or expert judgements.

The estimates of transport and effect factors of the different nutrient sources presented in this study are approximate, but they offer an alternative approach (compared to traditional calculations based on total annual amounts) to assess the relative importance of the various nutrient sources for eutrophication. In principle, the method can be used in the planning of water action plans without or with life cycle thinking. In industry, for example, the methods can be used for comparing and choosing between processes and materials in order to minimize environmental consequences. Authorities can use the assessment method for environmental action plans.

The method for calculating the contribution of nutrients to eutrophication is simple and provides only an approximate estimate of reality. Because there is no empirical test applicable for evaluation of the validity of results, the use of sensitivity and uncertainty analysis is important for their interpretation. In the case study, the uncertainty in the input variables was only determined as ranges by using expert assessments. A lack of scientific knowledge about uncertainty intervals for model variables means that the best way to use the method for the LCIA purposes in the current situation is to run different scenarios in which there are different data and assumptions for the calculation of source-dependent characterisation factors and a reference value. In future, it might be advantageous to determine relevant probability distributions (e.g. normal or log-normal) for input variables on the basis of experimental data in order to obtain more relevant uncertainty intervals. This would provide the possibilities to calculate confidence intervals for model results, which would help decision making because of smaller value ranges compared with the minimum and maximum values.

The method offers a bottom-up approach, in particular for creating an aquatic eutrophication model of Europe. It is easy to generate area-specific characterisation factors from sub-area-specific characterisation factors. The method offers an alternative approach compared with a top-down approach (c.f. Huijbregts and Seppälä 2001) in order to produce site-specific characterisation model on the European level. The weakness of the method presented in this paper is related to the accessibility of input data. However, it seems that this will not be a problem in future. This is due to the fact that directive 2000/60/EC of European Union requires that Member States determine 'target nutrient loads' of aquatic eutrophication in the catchments of each country (European Union 2000). It is expected that this work will produce more accurate bases for transport and effect factors of nutrients in inland and coastal waters in different European countries.

Acknowlegements. This article is based on work that was carried out at the Finnish Environment Institute on life cycle assessment. The authors acknowledge the contribution to the background data by Mr. Kaarle Kenttämies, Dr. Seppo Rekolainen, Dr. Petri Ekholm and Mr. Antero Nikander, who took part in the determination of coefficients. Mr. Michael Bailey revised the English of the manuscript.

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Received: August 27th, 2003
Accepted: January 16th, 2004
OnlineFirst: February 9th, 2004